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# *UK air quality showed clear improvement from 2015 to 2024 but breaching of targets remains very common*

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# UK air quality showed clear improvement from 2015 to 2024 but breaching of targets remains very common†

James Weber * and Helen F. Dacre

The impact of poor air quality (AQ) on public health has long been recognised and considerable efforts have been made to improve it across the UK. The UK has a far reaching AQ monitoring network and this study summarises the evolution of UK AQ over the period 2015–2024, focusing on the pollutants $NO_2$, $O_3$ and $PM_{2.5}$ and exploring their drivers. Concentrations of $NO_2$ and $PM_{2.5}$ exhibit robust negative trends across the whole country while concentrations of $O_3$ increase. Comparing 2015–2016 to 2023–2024, the median number of days per year for which DEFRA AQ sites breached the WHO 2021 target decreased from 136 to 40 (−70%) for $NO_2$ and from 60 to 22 (−63%) for $PM_{2.5}$. This trend was mirrored in other AQ monitoring networks and highlights that, while progress is being made, acceptable levels of AQ are yet to be reached. Over the same period, median $O_3$ exceedances increased from 7 to 14 days per year. Nationwide analysis of diurnal variation in the pollutants and the use of airmass back trajectory clustering and statistical modelling for three locations – Reading, Sheffield and Glasgow – suggests that local traffic plays a dominant role in $NO_2$ pollution, $PM_{2.5}$ is influenced more by long range transport and $O_3$ increases are being driven in part by decreases in $NO_2$. From an AQ policy perspective, this suggests continued focus on traffic emissions will reduce $NO_2$, (inter)national rather than local efforts are most critical for $PM_{2.5}$ improvements, and reductions to VOC emissions must accompany $NO_2$ if further $O_3$ increases are to be avoided.

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### Environmental significance

UK air quality (AQ) analysis is often done on a local (*e.g.* city-level) scale given the responsibility of local authorities to improve air quality. There is less focus on national level AQ trends, yet these are important because they cover a much greater population and differences across larger areas provide information as to the major sources of pollution (local emissions v. long range transport) and thus which policies will be effective. We find $NO_2$ and $PM_{2.5}$ concentrations decrease but breaching of AQ targets remains too frequent while $O_3$ increases. $NO_2$ reductions are linked to traffic while comparison of three distant locations suggests $PM_{2.5}$ is driven more by long range transport, implying national/international, not just local, policies are needed for its reduction.

## 1 Introduction

The World Health Organisation (WHO) estimates that 4.2 million deaths can be attributed to air quality (AQ) each year[1] and poor AQ is the largest modifiable global environmental public health risk.[2]

Poor AQ is generally defined in terms of the exposure to high concentrations of key pollutants which negatively affect human and/or vegetation health, including nitrogen dioxide ($NO_2$), ozone ($O_3$) and particulate matter smaller than 2.5 μm in diameter ($PM_{2.5}$).

$PM_{2.5}$ particles are small enough to reach deep into the lungs, and in some cases, the bloodstream with subsequent transport around the body and deposition in organs. Elevated levels of $PM_{2.5}$ are associated with a range of chronic health conditions including respiratory and neurological diseases.[3,4] Exposure to high concentrations of $NO_2$ and $O_3$ can also lead to serious health problems.[5,6]

Pollutant concentrations are determined by the balance of sources (local emissions, longer range transport into a region and chemical production in the atmosphere) and sinks (deposition to terrestrial or aqueous surfaces, chemical destruction and dispersion in the atmosphere). While poor AQ ultimately stems from emissions, both anthropogenic (*e.g.* nitric oxide (NO) from internal combustion) and natural (*e.g.* desert dust), meteorology also plays a major role. Stagnant, high pressure conditions can lead to the build up of pollutants from local sources. This infamously happened in the 1952 London Smog

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where subsidence associated with an anticyclone led to the formation of an atmospheric temperature inversion which prevented pollution from being mixed vertically through the atmosphere, effectively trapping pollutants close to their sources at the surface.[7] High wind speeds can disperse local pollution, improving AQ but, depending on their origin, can also transport pollution to a region from elsewhere. Precipitation can enhance loss of soluble pollutants while direct sunlight and higher temperatures drive enhanced photochemical $O_3$ production.[8]

The influence of meteorology means that it needs to be accounted for when assessing AQ. One way to do this is to use statistical techniques to model directly the impact of meteorology on AQ.[9] Another approach, adopted here, is to consider multiyear trends since, over such a period of time, most meteorological conditions will have been sampled at a frequency approximately representative of the longer climatology.

The importance of AQ has been enshrined in law in the United Kingdom for nearly 70 years[10] with legislation updated since.[11,12] Local authorities are required to monitor AQ in their region and develop air quality management areas (AQMA) in localities where government targets are unlikely to be met.[13,14] Since action on emissions is the predominant way policy can influence AQ at scale, several local authorities including Birmingham, Sheffield, Bath and London have enacted clean air[15] or low emission[16,17] zones, typically levying fines on certain vehicles when they enter. This study does not seek to evaluate the effectiveness of such zones; rather we examine nation-wide trends over the last decade.

Satellite observations over 2005–2015 of the UK found reductions in total column $NO_2$ and aerosol optical depth (AOD), a measure of particulate matter abundance, with $NO_2$ reductions largest in populated areas (~1–2% per year) while reductions in AOD (2.8–3.3% per year) were more spread out.[18] Long term (1992–2019) reductions in $NO_x = (NO + NO_2)$ have also been recorded at urban and rural surface sites.[19–21] Particulate matter concentrations have also declined over the last 30 years.[22] However, the difference in concentration between roadside and urban background concentrations is much smaller than that for $NO_2$, likely due to long range transport's larger contribution to particulate matter concentration,[23] a topic explored further in this study.

In contrast, $O_3$ has either changed negligibly or increased in recent decades.[18] Over the period 1990–2006, surface $O_3$ increased in rural (mean annual trend 0.28 $\mu g\,m^{-3}$ per year) and urban (0.79 $\mu g\,m^{-3}$ per year) sites while $NO_x$ concentrations also decreased in urban locations.[21] This was attributed to a combination of gradually increasing hemispheric $O_3$, seasonal increases due to regional production *via* reactions of volatile organic compounds (VOCs) with $NO_x$, and reduced local removal *via* $O_3$ + NO due to reducing $NO_x$ emissions. Diaz *et al.* (2020) also found consistent long-term increases in mean $O_3$ in rural and urban sites (1992–2019). However, maximum values of $O_3$, which typically occurred in May, exhibited decreases.

This study builds on prior work by documenting the change in the concentration of $NO_2$, $O_3$ and $PM_{2.5}$ over the last 10 years at more than 500 sites around the UK, compares UK trends to those in continental Europe, relates these concentrations changes to the changes in the frequency with which AQ targets are breached, and decomposes pollutant concentrations by hour of day, day of week and location (urban, rural *etc.*) to determine what drives their spatial and temporal variability.

Finally, three locations, Reading, Sheffield and Glasgow, located geographically in the south, middle and north of the UK respectively, are considered in detail. AQ, meteorological and airmass back trajectory data are used alongside statistical modelling to compare the major of drivers of $NO_2$, $O_3$ and $PM_{2.5}$ variability across the locations and the implications for effective AQ policy are discussed.

## 2 Data

Hourly UK air quality data were sourced from the UK Automatic Urban and Rural Network (AURN), the Air Quality England network (AQE), the Scottish Air Quality Network (SAQN), the Welsh Air Quality Network (WAQN), the Northern Ireland Air Quality Network (NI), locally managed air quality networks and the KCL/Imperial network. Hourly meteorological data for Reading were sourced from the Reading University Atmospheric Observatory, for Sheffield from the Sheffield Airvivo site and for Glasgow from worldmet package. See the data availability section for more information. Unless otherwise stated, trends were calculated using the Theil-Sen function within Openair on monthly mean concentrations.

## 3 Results

### 3.1 2015–2024 trends

Trends in the concentrations of $NO_2$, $PM_{2.5}$ and $O_3$ were calculated for every AURN, AQE, SAQN, WAQN, NI, Local and KCL/Imperial site for the 10-year period 2015–2024. The concentration of $NO_2$ at every qualifying‡ AURN site exhibited a negative trend (median decadal change of −35%) (Fig. 1 and S1-2†) with Urban Traffic sites typically having the most negative values. This reduction in $NO_2$ was also exhibited in sites from the AQE (England, Fig. S3†), SAQN (Scotland, Fig. S4†), Wales (WAQN, Fig. S5†), Northern Ireland (NI, Fig. S5†), Local (all regions, Fig. S6†) and KCL/Imperial (one site with positive trend, Fig. S7†) networks. Combined this comprises 503 sites (117 AURN, 386 other) and the negative trend follows that observed over the rest of Europe (Fig. 2).

Of the 199 qualifying sites for $PM_{2.5}$ (72 AURN, 127 other), all but 10 exhibited decreasing trends in concentration (median decadal change −30% for AURN) (Fig. 1, S1, S2, S7 and S8†), also largely in line with the European wide decrease (Fig. 2). An obvious feature of both the absolute (Fig. 1 and S1†) and fractional (Fig. S2†) trend plots is that there is not a clear correlation between the trends in $NO_2$ and $PM_{2.5}$ ($R^2 = 0.06$ for absolute trends). In other words, a site with a large reduction in $NO_2$ does not necessarily have a large reduction in $PM_{2.5}$. If the sites with the largest $NO_2$ reductions also exhibited the largest $PM_{2.5}$

‡ Minimum 75% data coverage, from 2015 to at least 2023, trend statistically significant at 95% confidence level.


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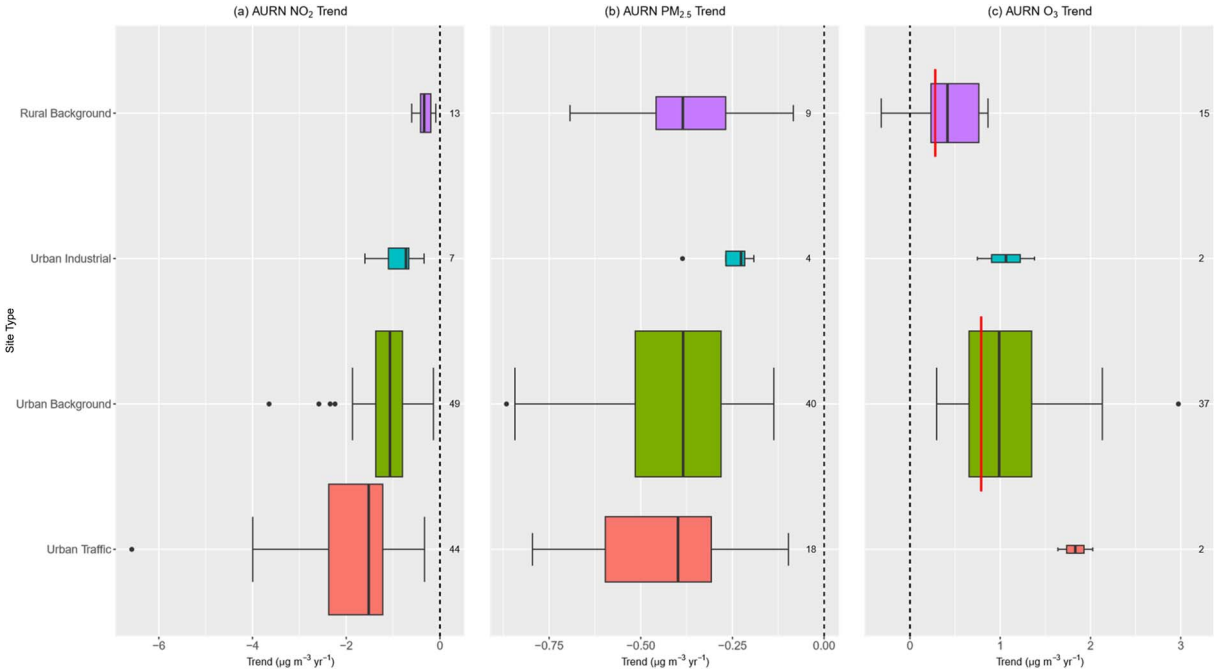


Fig. 1 Deseasonalised trend over 2015–2024 from AURN sites of (a) $NO_2$, (b) $PM_{2.5}$ and (c) $O_3$. Only sites with at least 75% coverage of the time period starting in 2015, running to at least 2023, and where trends are significant at the 95% level are included. The number of qualifying locations for each site type is given by small numbers on the right of each subplot and the height of each boxplot is proportional to the number of qualifying sites. Red lines in (c) show the mean annual trend in $O_3$ at the urban (1993–2006) and rural (1990–2006) sites considered in Jenkin (2008).[24]

reductions, it could be inferred that the pollutants were largely coming from the same source but, as this is not the case, Fig. 1 presents strong evidence that $PM_{2.5}$ and $NO_2$ are driven by different factors, a finding in line with prior work[23] and subject of further discussion in Section 3.4.1.

In contrast to the decreasing concentrations of $NO_2$ and $PM_{2.5}$, $O_3$ concentrations exhibited increases in 115 of the 121 sites considered (56 AURN, 65 other) (Fig. 1, S7 and S9†) with a median decadal increase of 17% at AURN sites. This increase is in line with positive trends exhibited over much of Northern Europe over the last 10 years (Fig. 2) and qualitatively in agreement with prior UK-focused studies.[19,21]

### 3.2 Exceedances

While trends in concentration are useful, changes to the frequency with which AQ targets are exceeded is more informative from a public health perspective. (For clarity, it should be noted that an exceedance of a target is a negative event in the context of AQ). Indeed, while the AURN site on London's Marylebone Road showed the most negative trend in $NO_2$ concentration (Fig. S10†), it still had the greatest number of exceedances in 2023–2024 (~290 days per year) of the 24-hour WHO AQ 2021 target.[25] Following decreases in their concentrations, the number of days where these targets were exceeded decreased for $NO_2$ and $PM_{2.5}$ for all qualifying AURN, AQE, SAQN, WAQN, NI and Local network sites (Table 1, Fig. 3 and S10–S15†). Considering sites for which there are data for at least 300 days for 2015, 2016, 2023 and 2024 (to ensure consistency), the annual median for $NO_2$ exceedances decreased by 70% at AURN sites and 61% for other sites (AURN, AQE, SAQN, WAQN, NI and Local network) while for $PM_{2.5}$ exceedances dropped by 63% (AURN) and 68% (other). Despite this progress (also exhibited by sites with lower data coverage, Fig. S16†), very few sites met the WHO AQ 2021 targets for $NO_2$ and $PM_{2.5}$ of no more than 3–4 exceedances per year, even in 2023–2024.

In line with the increasing $O_3$ concentrations, the number of exceedances of WHO 2021 daily maximum 8-hour average (MDA8) $O_3$ target[25] increased from 2015–2016 to 2023–2024 (Table 1, Fig. 3, S10 and S17†). We note this is the opposite finding to Diaz *et al.* (2020),[19] possibly because they used a different AQ target (number of hours per year for which $O_3$ exceeded 50 ppbv ≈ 100 µg $m^{-3}$). However, the median number of MDA8 exceedances is lower than for $NO_2$ or $PM_{2.5}$ (Table 1) although this may change if $NO_2$ or $PM_{2.5}$ continue to decrease, suggesting the focus on AQ policies may need to shift to place greater consideration on $O_3$ in coming decades.

**3.2.1 $NO_2$ and $O_3$ Tradeoffs.** In most of the urban and semi-urban areas of the UK, local $O_3$ production is volatile organic compound (VOC) limited, such that increases in $NO_x$ will reduce $O_3$.[24] Therefore, as $NO_2$ concentrations decrease (Fig. 1, 2 and S1-7†), $O_3$ concentrations are likely to increase, contributing to the increasing frequency with which $O_3$ AQ targets are exceeded. For example, while only a correlation, it is noted that the annual mean concentration of $NO_2$ on days when $O_3$ breached the WHO 2021 AQ target has decreased over the last decade at all qualifying AURN sites (Fig S18†). From a public

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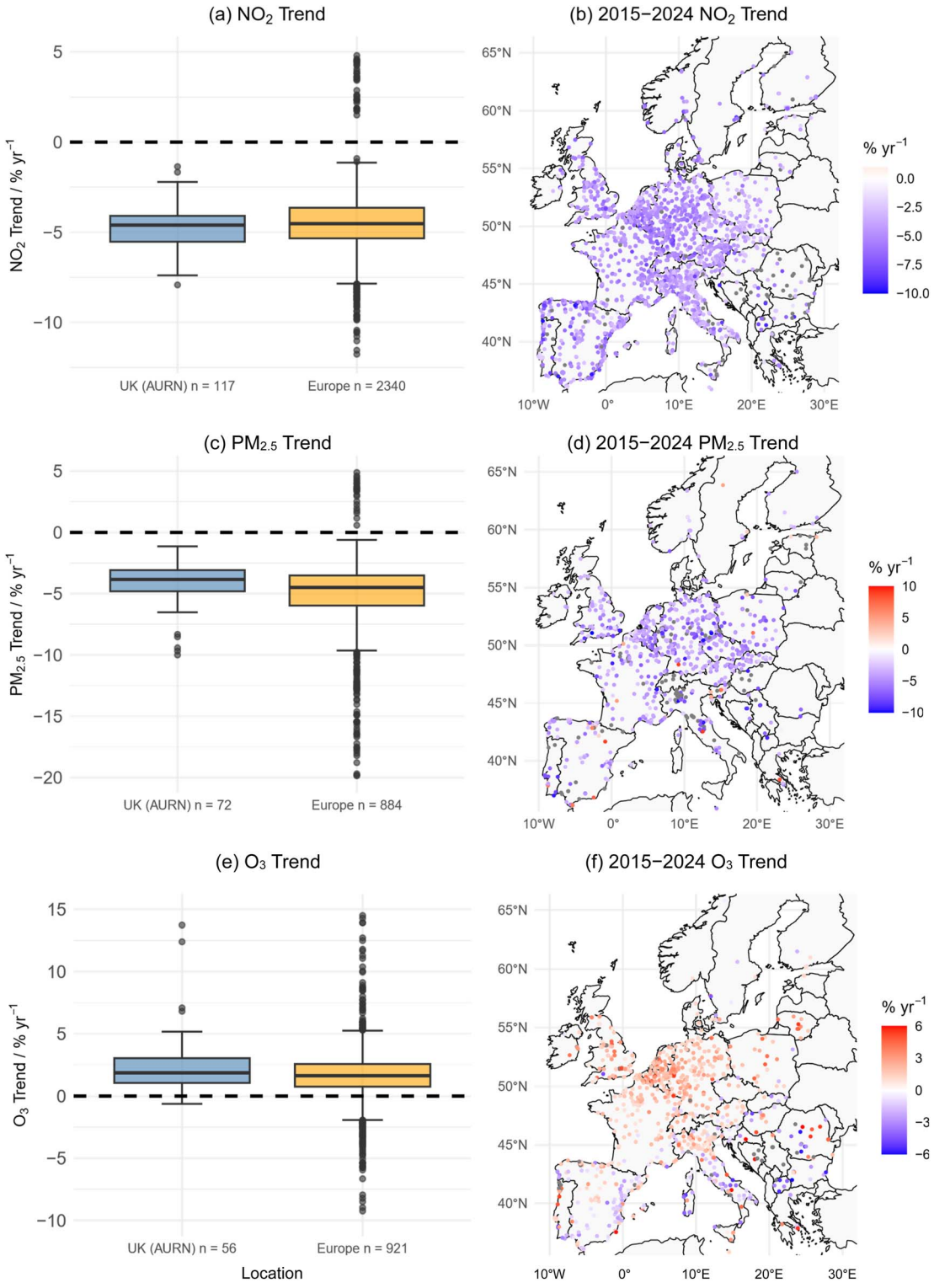


Fig. 2 Deseasonalised percentage trends over 2015–2024 of (a and b) $NO_2$, (c and d) $O_3$ and (c and f) $PM_{2.5}$. Only sites where data exist from 2015, there is at least 75% data coverage and trends are statistically significant at 95% confidence are included. UK sites in (a, c and e) are from AURN sources.

health perspective, one way to assess this trade-off is to compare the change in frequency with which $NO_2$ and $O_3$ AQ targets are being breached. Comparing 2015–2016 with 2023–2024, the increase in the number of days when the $O_3$ target was exceeded is vastly outweighed by the decrease in the number of days when the $NO_2$ target was exceeded (Fig. 4 and S19†). In other words, there has been a net positive benefit (on an exceedance basis) to AQ from the reduction in $NO_2$ over the last 10 years. The


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Table 1 Annual median [25th percentile, 75th percentile] number of days which qualifying sites exceeded WHO AQ 2021 targets. To ensure a consistent comparison, qualifying sites are defined as those for which there are data for at least 300 days in all four years considered. For each site, the mean number of exceedances for 2015–2016 and for 2023–2024 were taken and then a median was taken across all sites. Other refers to AQE, SAQN, WAQN, NI and local network sites

| Pollutant | Sites | 2015–2016 | 2023–2024 |
|---|---|---|---|
| $NO_2$ (AURN) | 68 | 136 [82, 230] | 40 [22, 75] |
| $NO_2$ (other) | 145 | 225 [172, 308] | 88 [50, 162] |
| $PM_{2.5}$ (AURN) | 40 | 60 [46, 70] | 22 [20, 30] |
| $PM_{2.5}$ (other) | 18 | 47 [27, 56] | 15 [8, 26] |
| $O_3$ (AURN) | 53 | 7 [5, 10] | 14 [9, 16] |
| $O_3$ (other) | 25 | 6 [2, 7] | 9 [6, 12] |

response of $O_3$ to changes in $NO_x$ (emissions) is complex and also depends on temperature and the concentrations of VOCs and carbon monoxide (CO), which acts in a similar way to VOCs in the production of $O_3$ but is often grouped separately.

If $NO_2$ continues to decrease, more regions may move into a $NO_x$-limited regime where reductions to $NO_x$ will drive $O_3$ decreases (a "win–win" for AQ). However, changes to anthropogenic VOC/CO emissions and rising temperatures (which tend to lead to elevated $O_3$ (ref. 26)), may distort this and lead to, in the short term at least, enhanced $O_3$ sensitivity to further $NO_2$ reductions.

### 3.3 Diurnal variations and year-on-year changes

Analysis of the diurnal variation of $NO_2$, $O_3$ and $PM_{2.5}$ and comparison of cycles from urban and rural sites is informative in determining their major drivers. For each site type and year, a diurnal cycle was constructed by first calculating the mean diurnal cycle for each location with at least 80% data coverage for all years 2015–2024 and then taking a mean over all sites §.

$NO_2$ concentrations show pronounced increases in the morning and late afternoon/early evening during the week, particularly at urban traffic sites and, to a lesser extent, urban background sites (Fig. 5). Weekend profiles exhibit a much smaller or even non-existent morning peak and the evening peak typically occurs slightly later than during the week. The contrast between urban week and weekend profiles and the much lower concentrations observed in rural sites points to local traffic emissions as the major driver of $NO_2$. As in Fig. 1 and 2, $NO_2$ concentrations show clear reductions year-on-year to the extent that $NO_2$ at urban traffic sites on weekdays in 2023 and 2024 was lower than in 2020, despite the significant reductions in traffic in 2020 during COVID19 lockdowns.[27]

By contrast, $O_3$ concentrations show year-on-year increases and are higher at rural than urban sites. Concentrations reach a minimum at urban background sites around 6–7 am GMT, coinciding with the start of the morning rush hour and reduced sunlight. This minimum is more pronounced on weekdays than at the weekend, pointing towards suppression of $O_3$ by $NO_x$ from traffic as a driver. Peak $O_3$ concentrations occur in the early afternoon (~2 pm). As tropospheric $O_3$ has a lifetime of ~20–30 days (*c.f.* $NO_2$'s lifetime of ~hours), the diurnal cycle in $O_3$ concentrations is driven by the entrainment of $O_3$ from the free troposphere into the growing boundary layer and deposition to the surface at night, rather than local diurnal $O_3$ production. This entrainment of free tropospheric air complicates the attribution of the drivers of the year-on-year increase in surface $O_3$ by adding another $O_3$ source.

Quantifying the relative importance of local and entrained $O_3$ is challenging, particularly as $O_3$ from northern continental Europe (which will contribute to the entrained free tropospheric $O_3$ experienced in the UK) is also increasing (Fig. 2). Nevertheless, several factors point to reduction in local $NO_2$ (and decrease in $NO_x$-driven suppression) in being influential and thus worthy of policymakers' attention. For example, for all site types, $O_3$ is higher at the weekend than on weekdays, despite both experiencing free tropospheric $O_3$ of similar concentrations, suggesting this difference is driven more by a reduction in $NO_x$-driven suppression. The elevated $O_3$ at rural sites compared to urban sites, the morning rush hour weekday minimum and the anomalously high concentrations in 2020 also support this and provide further evidence that, at the very least, most urban sites in the UK are in VOC-limited regimes, in agreement with prior studies.[9] UK wide anthropogenic emissions of CO are predicted to have decreased by 13.4% between 2015 and 2024 (ref. 28) (Fig. S20†) and surface concentrations of CO are decreasing (albeit over a much more limited sensor network, Fig. S21†). However, emissions of NMVOCs, which can produce more $O_3$ per molecule than CO can when they react in the atmosphere, have decreased by less than 1% over that time and both CO and NMVOC emission reductions are outstripped by the 20% decrease in $NO_x$ emissions. The response of $O_3$ here suggests that the ongoing efforts to reduce $NO_2$ must be accompanied by enhanced efforts to reduce anthropogenic VOC and CO emissions if further increases in $O_3$ are to be avoided. Rising UK temperatures[29,30] and proposals to increase forest cover[31] will likely increase UK biogenic VOCs emissions. These have been lower than anthropogenic VOC emissions over the last two decades (by a factor varying between ~2.5 and ~10 intra-annually over 2015–2020) (Fig. S22†) but future increases will fuel $O_3$ production. This compounds the importance of reducing anthropogenic VOC emissions still further.

$PM_{2.5}$ diurnal variations in urban areas also exhibit morning rush hour peaks but, in contrast to $NO_2$, the evening peak tends to see higher concentrations and also occur later (~7–8 pm GMT *vs.* 4–5 pm GMT for $NO_2$ in weekday urban traffic sites) suggesting a possible greater contribution from domestic heating, the impact of a shallower nighttime boundary layer and lower sensitivity to local emissions. Furthermore, there is also less of a difference between urban traffic and urban background concentrations and between weekday and weekend profiles. All this points to traffic being a relatively less important source of $PM_{2.5}$ than $NO_2$, aligning with the trend analysis in Fig. 1.

§ Various tests were performed to explore the implications of different types of averaging (*i.e.* the mean of the means, median of the medians, median of the means, *etc.*) but no substantive difference to the final result were found.

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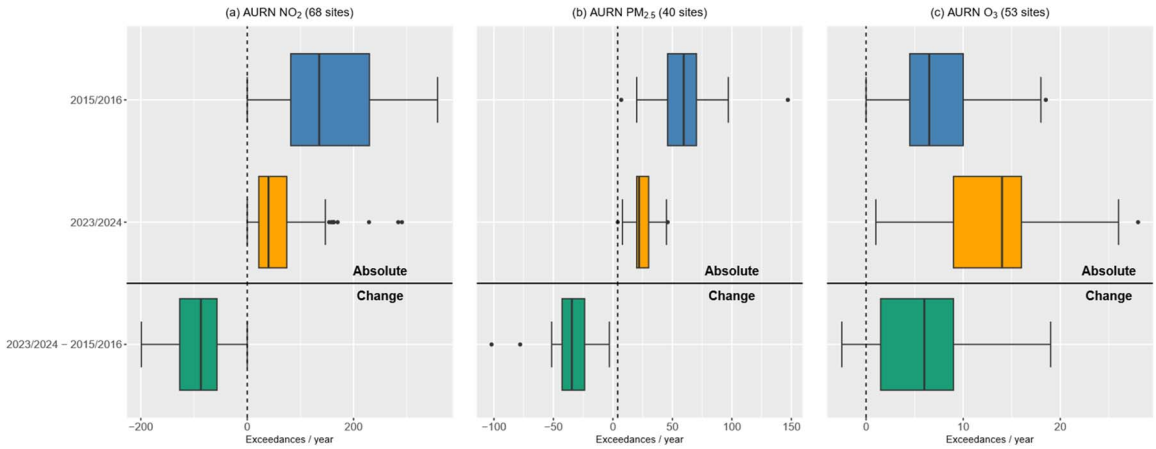


Fig. 3 Mean number of exceedances of WHO 2021 air quality targets for 2015–2016 and 2023–2024 (upper section) and site-specific change in exceedances (lower section) for AURN measurements of (a) $NO_2$, (b) $PM_{2.5}$ and (c) MDA8 $O_3$. For each site, exceedances are only given if there are data for at least 300 days in all four years. Exceedances should occur on no more than 3–4 days per year.

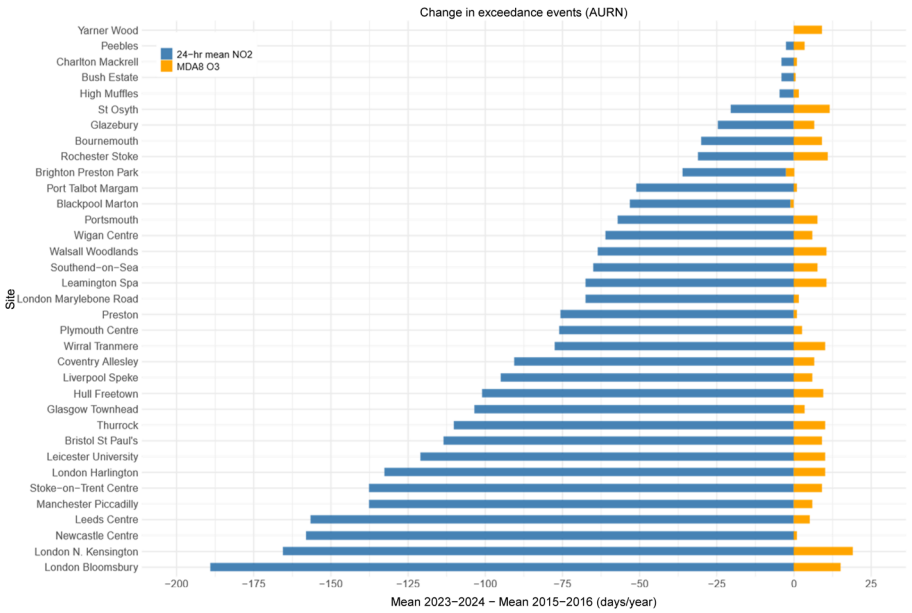


Fig. 4 Change in mean number of exceedances between 2015–2016 and 2023–2024 for AURN sites with measurements of both $NO_2$ and MDA8 $O_3$. Criteria for calculating exceedance is the same as in Fig. 3.

These trends in $NO_2$, $O_3$ and $PM_{2.5}$ are also seen when the year-on-year evolution of the seasonal cycles from AURN sites are examined (Fig S23†). $NO_2$ exhibits the highest concentrations in winter months, in line with shallower boundary layers trapping local emissions closer to the surface, but also a clear year-on-year on decrease. Conversely, $O_3$ peaks in April/May, in part due to elevated temperatures promoting deeper boundary layers and free tropospheric air entrainment, and increased consistently over the last decade. $PM_{2.5}$ exhibits a more muted seasonal cycle, likely due to the greater role of long-range transport which can occur throughout the year, but steady decreases over the period.

The extent to which $NO_2$ and $PM_{2.5}$ concentrations may continue to change is not the main focus of this study, but some tentative predictions may be derived from Fig. 5 and the year-on-year difference plots (Fig. S24–S26†). Aside from the anomalous result of 2020, urban $NO_2$ concentrations exhibit consistent year-on-year decreases; indeed the 2023 to 2024 decrease was larger than that between 2022 and 2023. This would suggest that a "plateauing" of $NO_2$ concentrations has not yet been reached (*i.e.* ongoing trends such as the replacement of older vehicles in the fleet with newer, cleaner ones will deliver continued $NO_2$ concentration reductions¶) but this can only be confirmed by further monitoring in real time and accurate modelling. $PM_{2.5}$ presents a more complex picture – the

¶ This is neither an endorsement nor criticism of clean air/low emission zones.


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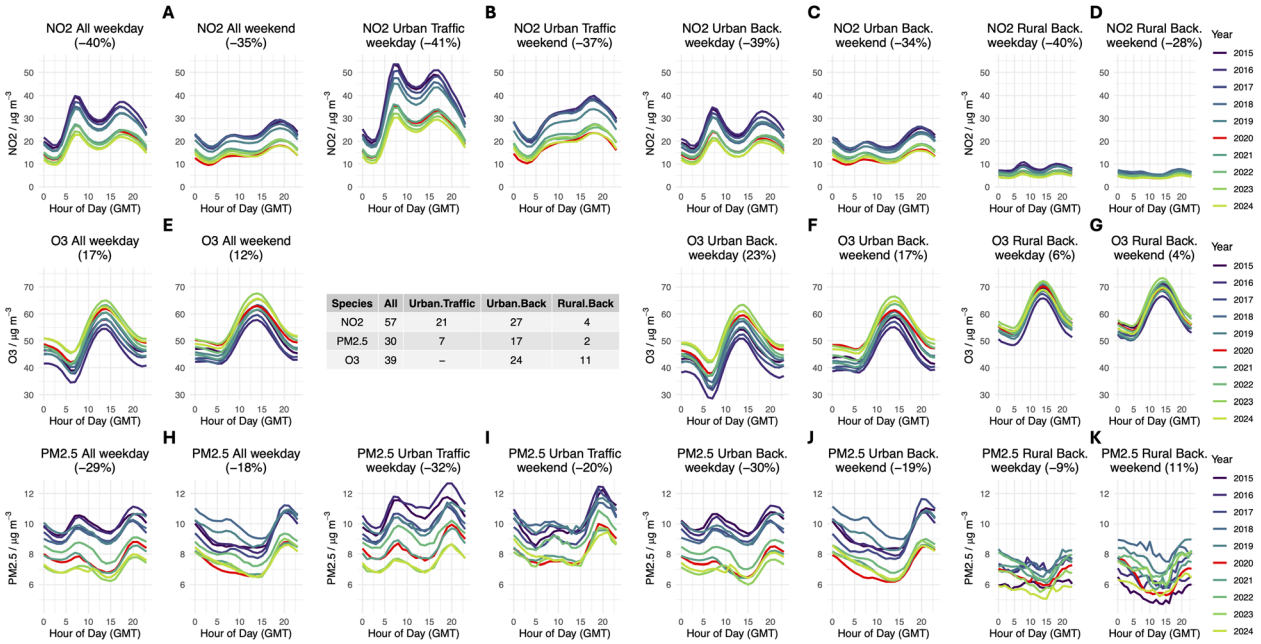


| Species | All | Urban.Traffic | Urban.Back | Rural.Back |
|---|---|---|---|---|
| NO2 | 57 | 21 | 27 | 4 |
| PM2.5 | 30 | 7 | 17 | 2 |
| O3 | 39 | – | 24 | 11 |

Fig. 5 Weekday and weekend diurnal cycles for (A–D) $NO_2$, (E,F,G) $O_3$ and (H–K) $PM_{2.5}$ from AURN sites. Table shows number of qualifying sites for each species and site type. Mean diurnal cycles constructed from mean of sites with at least 80% coverage in each year 2015–2024. No Urban Traffic sites satisfied the criteria for $O_3$. Parenthetical numbers show change from 2015–2016 daily mean to 2023–2024 daily mean. "All" refers to all types of site.

decrease from 2022 to 2023 was greater than that between 2023 and 2024 – and this is likely in part to be due to the more complex array of sources and enhanced dependence on long range transport (see Section 3.4.1). Meanwhile, future trends in $O_3$, as previously discussed, are likely to be positive as $NO_x$ is reduced in chemical regimes where $NO_x$ suppresses $O_3$ (although, as previously mentioned, it is possible some regions will eventually tip into $NO_x$-limited regimes where further $NO_x$ reductions will reduce $O_3$).

### 3.4 Case study – comparing drivers of $NO_2$, $O_3$ and $PM_{2.5}$

In this final section, attention is paid to the variation of $NO_2$, $O_3$ and $PM_{2.5}$ measured at AURN urban background monitors in locations in the south (Reading), middle (Sheffield) and north (Glasgow) of the UK. Fig. 6A–C show the clustered 96-hour back trajectories for airmasses arriving at Reading, Sheffield and Glasgow. The majority of airmasses arriving at all three locations come from the west or northwest with smaller fractions from continental Europe. Fig. 6D–L decomposes $NO_2$, $O_3$ and $PM_{2.5}$ to show how they vary by airmass cluster, hour of day and month of year for 7 years (August 2017 to July 2024).

$NO_2$ (Fig. 6D–F) exhibits the highest concentrations in winter months, likely due in part to shallower atmospheric boundary layers reducing pollution transport away from the surface, and the diurnal variation of Fig. 5 is also visible. While airmasses coming from the Atlantic are associated with lower $NO_2$ (C1–C4), no one cluster is associated with much higher concentrations. By contrast, $PM_{2.5}$ is highest in each location when the air has come from the south-east/European continent (C7 for Reading and Sheffield, C8 for Glasgow).

$O_3$ exhibits maximum concentrations in the mid to late afternoon in the summer months, the period with highest temperatures and boundary layer height. Airmasses coming from northwest Europe are associated with the highest probability of $O_3$ exceeding AQ targets, followed by southerly airmasses.

The similarity in the $PM_{2.5}$ dependence on airmass origin for the three locations points towards long range transport as the dominant driver. Airmasses from the southeast are likely to contain a mixture of anthropogenic pollution (including from agriculture) from continental Europe and southern England (particularly for Sheffield and Glasgow) as well as Saharan dust on some occasions. This finding is broadly in line with prior modelling studies.[32–34]

This dependence is also reflected, to some extent, when the variation of exceedance probability with cluster is examined (Fig. 5O and P). The probability of $PM_{2.5}$ breaching the WHO AQ 2021 target is more than 15% only when the airmass originates from the South or East for Reading (C7, C8), Sheffield (C7, C8), and Glasgow (C6, C7). By contrast, the probability of $NO_2$ exceedance in Reading is high for several "clean" air masses origins as well as C7 and C8.

**3.4.1 Statistical modelling of pollutant concentrations.** To isolate the influence of different factors on $NO_2$, $O_3$ and $PM_{2.5}$ in the three locations, statistical models of each of pollutant in each location were built using the Deweather software.[35] 96-

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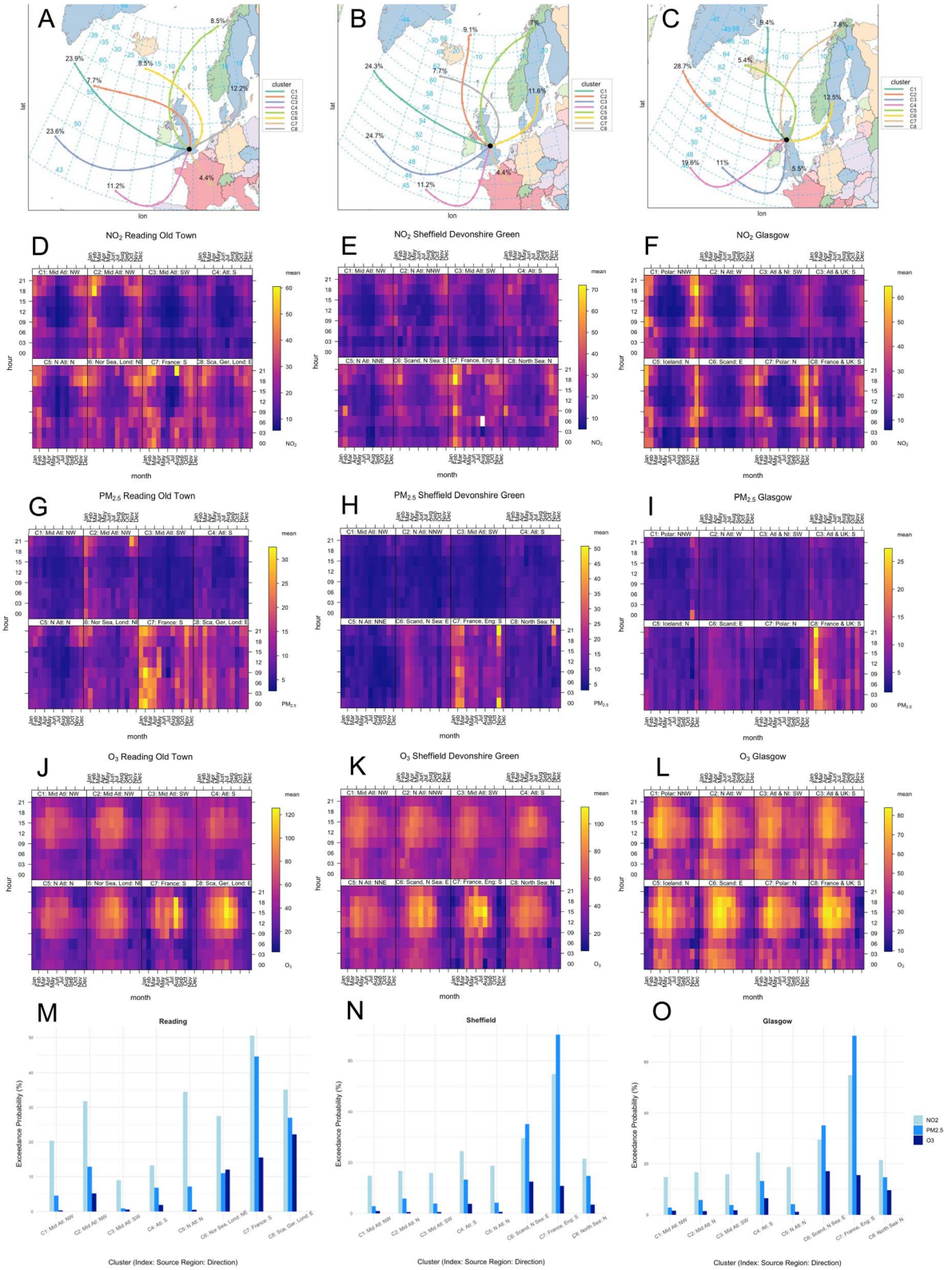


Fig. 6 (A–C) Air mass back trajectory clusters for Reading (A), Sheffield (B) and Glasgow (C). Mean concentrations of $NO_2$ (D–F), $PM_{2.5}$ (G–I) and $O_3$ (J–L) by hour of day and month of year separated by air mass cluster and WHO 2021 AQ target exceedance probability (M–O) for Reading, Sheffield and Glasgow (Aug 2017–July 2024). Percentage values in A–C show fraction of back trajectories which fall into each cluster. D–L have units of $\mu g\ m^{-3}$.


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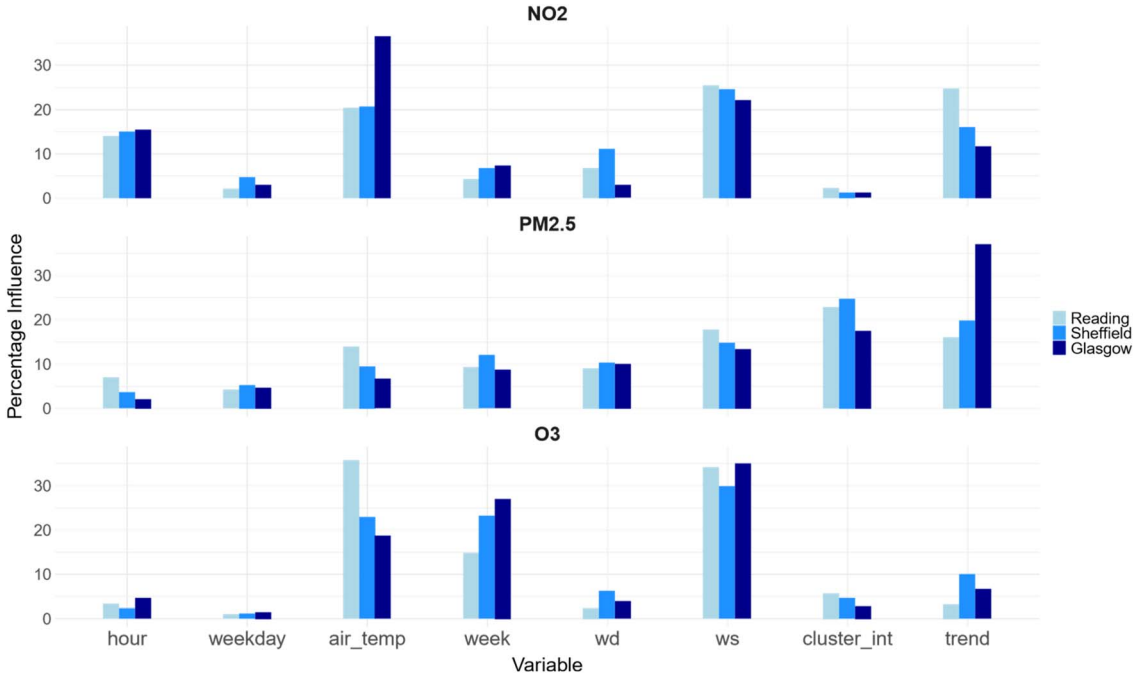


Fig. 7 Influence of explanatory variables and trend used by the deweather package when building statistical models of $NO_2$, $PM_{2.5}$ and $O_3$ for Reading, Sheffield and Glasgow. Variable "hour" refers to hour of day, "weekday" to day of week, "week" to week of year, "wd" to wind direction, "ws" to wind speed and "cluster_int" refers to the air mass back trajectory clusters in Fig. 6. Full PDs shown in Fig. S27–S29.†

hour back trajectories centred on a receptor altitude of ~250 m were generated using HYSPLIT (see data availability section) for each location for August 2017 to July 2024 inclusive on a 3-hourly basis. (Varying the receptor height in sensitivity tests did not materially influence the clusters.) These back trajectories were categorised into one of 8 clusters (Fig. 6A–C), generated using the angle-based clustering algorithm in Openair.[36] These 3-hourly cluster data were merged with contemporaneous meteorological data and used as input for the Deweather model. Specifically, explanatory variables of wind speed, wind direction, air temperature, air mass origin (discretised using the clusters), hour of day, day of week and week of year were used. From these models, each variable's partial dependency (PD), generated by changing one variable while holding all others at their mean value, was calculated. The relative influence of each variable, along with the trend which can be considered as everything not captured by the model's explanatory variables (for example, changes in emissions), on the pollutants is shown in Fig. 7 while the full response of each pollutant to each explanatory variable and the trend is plotted in Fig. S27–S29.†

All locations are consistent in having the airmass origin as the most important explanatory variable for $PM_{2.5}$. Wind speed is the second most influential explanatory variable in each location with low speeds associated with high $PM_{2.5}$. Such conditions are conducive to the build up of $PM_{2.5}$ from local emissions and anticyclones over northern Europe whose easterlies along the south side drive long range transport of $PM_{2.5}$ from the continent.[34] All locations exhibit a negative trend, suggesting emissions, local and further afield, are decreasing.

For $NO_2$, air temperature and wind speed are the most important explanatory variables. Increasing wind speed and temperature, conditions associated with enhanced turbulence with a deeper boundary layer and thus dispersion of pollution away from the surface, lead to lower modelled $NO_2$. $NO_2$ also displays a much greater dependence on hour of day than $O_3$ or $PM_{2.5}$ in all locations, in line with Fig. 5 while airmass origin (*i.e.* long range transport) accounts for less than 3% of the variability in $NO_2$.

$O_3$ is also influenced substantially by temperature and wind speed but in the opposite way to $NO_2$. Modelled $O_3$ increases with temperature and wind speed, indicative of enhanced production of $O_3$ and mixing of $O_3$-rich free tropospheric air into the boundary layer. Airmass origin is less important for $O_3$ when all hours are considered (accounting for 3–6% of variability), but more influential in the context of exceedance probability (Fig. 6O and P), particularly in Reading. When combined with the UK-wide analysis in Section 3.3, this suggests that while year-on-year increases are being driven by $NO_x$ reductions, $O_3$-rich continental air can contribute to elevating concentrations to the point where they breach AQ targets. Since free tropospheric $O_3$ is largely beyond the influence of local authorities, these findings highlight of the importance of local and regional efforts to reduce $O_3$ production by careful management of its precursor emissions: $NO_x$ and VOCs/CO.

## 4 Conclusions and implications for AQ policy

Effective policy must be informed by, among other things, a comprehensive and rigorous summary of the current state of

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affairs. This study highlights the progress the UK has made on AQ in the last decade, but also the significant amount of work remaining to bring AQ to acceptable levels and the potential unintended consequences of overly simplistic policies. The robust, UK-wide reduction in $NO_2$ contains strong signals that it is driven by changes to traffic emissions and, as such, offers hope that continuing efforts to reduce traffic emissions will lead to further AQ improvements. While $PM_{2.5}$ also exhibits some dependence on traffic/local emissions, the larger dependence on airmass origin (relative to $NO_2$), implies a greater role in regional and international transport. This means that while local $PM_{2.5}$ emission reductions have a modest contribution to make, effective policies to reduce $PM_{2.5}$ will require national and international efforts rather than just delegation to regional authorities.

In contrast to improvements in $NO_2$ and $PM_{2.5}$, $O_3$ pollution is worsening, likely due to local reductions in $NO_2$. While the benefits of reductions to $NO_2$ have hitherto outweighed the hazards posed by elevated $O_3$ (in terms of changes to frequency with which the WHO AQ targets are breached), this may not always be the case, particularly in a warming world where $NO_2$ is cut further. Effective policy at local scale is possible but this should not target $NO_2$ in isolation: simultaneous reductions in VOCs/CO must also be pursued to temper $O_3$ increases.

One area not explored in this study is compositional analysis of $PM_{2.5}$. While there are far fewer sites which can conduct this measurement compared to those which measure $PM_{2.5}$ concentrations, compositional analysis provides substantial information as to the origin of the particulate matter (and its toxicity[37]) and can thus inform effective mitigation.

Finally, this study, along with the vast majority of literature, focuses on outdoor AQ whereas most humans spend the majority of their lives inside. Indoor pollutant levels differ from those outdoors[38] and refining the understanding of the causes and thus possible mitigatory actions of poor indoor AQ, alongside outdoor AQ, should be viewed as a priority.

## Data availability

All AQ and meteorological data used in this study can be accessed using the Openair R package (last accessed 22nd March 2025), specifically the importUKAQ, importKCL, importEurope and importNOAA (side code 031400-99999 for Glasgow) functions, the Reading University Atmospheric Observatory (last accessed 22nd March 2025) or the Sheffield Airvivo site. The 96-hour back trajectories centered on Reading, Sheffield and Glasgow were generated using the HYSPLIT software.[39–42] Unless stated, all trends were calculated using the Theil–Sen function within Openair. The partial dependency analysis used the deweather package developed by David Carlsaw and colleagues at the University of York.[35] All analysis code, meteorological data from Reading and Sheffield, and back trajectory files for Reading, Sheffield and Glasgow have been uploaded to a publicly accessible Zenodo repository (**https://doi.org/10.5281/zenodo.15205925**) along with the adequate documentation.

## Author contributions

JW planned the study and extracted and analysed the data. JW and HFD discussed the analysis and both contributed to the writing of the manuscript.

## Conflicts of interest

There are no conflicts to declare.

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## Notes and references

1 World Health Organization, Types of Pollutants environment-climate-change-and-health/air-quality-and-health/health-impacts/types-of-pollutants, accessed**https://www.who.int/teams/**: 2025-04-09,.
2 E. McDuffie, R. Martin, H. Yin and M. Brauer, *Research Reports: Health Effects Institute*, 2021, p. 210.
3 T. J. Grahame, R. Klemm and R. B. Schlesinger, *J. Air Waste Manage. Assoc.*, 2014, **64**, 620–660.
4 P. J. Landrigan, R. Fuller, N. J. Acosta, O. Adeyi, R. Arnold, A. B. Baldé, R. Bertollini, S. Bose-O'Reilly, J. I. Boufford, P. N. Breysse, *et al.*, *Lancet*, 2018, **391**, 462–512.
5 U.S. Environmental Protection Agency, *Integrated Science Assessment (ISA) for Oxides of Nitrogen – Health Criteria*, U.S. Environmental Protection Agency Technical Report EPA/600/R15/068, 2016.
6 B. Chen and H. Kan, *Environ. Health Prev. Med.*, 2008, **13**, 94–101.
7 E. Wilkins, *J. R. Sanit. Inst.*, 1954, **74**, 1–21.
8 J. Seinfeld and S. Pandis, *Atmospheric Chemistry and Physics: from Air Pollution to Climate Change*, 1998, p. 1326.
9 S. K. Grange, J. D. Lee, W. S. Drysdale, A. C. Lewis, C. Hueglin, L. Emmenegger and D. C. Carslaw, *Atmos. Chem. Phys.*, 2021, **21**, 4169–4185.
10 United Kingdom, Clean Air Act 1956, 1956, available at: **https://www.legislation.gov.uk/ukpga/Eliz2/4-5/52/enacted**, accessed 23 Mar. 2025.
11 United Kingdom, Clean Air Act 1968, 1968, available at: **https://www.legislation.gov.uk/ukpga/1968/62/enacted**, accessed 23 Mar. 2025.
12 United Kingdom, Clean Air Act 1993, 1993, available at: **https://www.legislation.gov.uk/ukpga/1993/11/enacted**, accessed 23 Mar. 2025.
13 United Kingdom, Environment Act 1995, 1995, available at: **https://www.legislation.gov.uk/ukpga/1995/25/enacted**, accessed 23 Mar. 2025.
14 L. Smith, *Local Government Air Quality Responsibilities, UK Parliament Technical Report CBP-8804*, 2025.


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15 Department for Environment, Food & Rural Affairs (DEFRA), Driving in a Clean Air Zone, UK Government Guidance, 2025, **https://www.gov.uk/guidance/driving-in-a-clean-air-zone**, accessed: 23 Mar. 2025.
16 Transport for London (TfL), Ultra Low Emission Zone (ULEZ), Transport for London Website, 2025, **https://tfl.gov.uk/modes/driving/ultra-low-emission-zone**, accessed: 23 Mar. 2025.
17 Transport Scotland, About Low Emission Zones in Scotland, Low Emission Zones Scotland, 2025, **https://lowemissionzones.scot/about**, accessed: 23 Mar. 2025.
18 R. J. Pope, S. R. Arnold, M. P. Chipperfield, B. G. Latter, R. Siddans and B. J. Kerridge, *Atmos. Sci. Lett.*, 2018, **19**, e817.
19 F. M. R. Diaz, M. A. H. Khan, B. M. A. Shallcross, E. D. G. Shallcross, U. Vogt and D. E. Shallcross, *Atmosphere*, 2020, **11**(5), 534.
20 Department for Environment, Food & Rural Affairs, Air Quality Statistics in the UK, 1987 to 2023 – Nitrogen Dioxide (NO), **https://www.gov.uk/government/statistics/air-quality-statistics/ntrogen-dioxide**, updated 18 March 2025. Crown copyright. Licensed under the Open Government Licence v3.0.
21 M. E. Jenkin, *Atmos. Environ.*, 2008, **42**, 5434–5445.
22 Department for Environment, Food & Rural Affairs, Air Quality Statistics in the UK, 1987 to 2023 – Particulate Matter (PM10/PM2.5), **https://www.gov.uk/government/statistics/air-quality-statistics/concentrations-of-particulate-matter-pm10-and-pm25**, updated 30 April 2024. Crown copyright. Licensed under the Open Government Licence v3.0.
23 R. M. Harrison, D. Laxen, S. Moorcroft and K. Laxen, *Atmos. Environ.*, 2012, **46**, 115–124.
24 M. E. Jenkin and K. C. Clemitshaw, *Atmos. Environ.*, 2000, **34**, 2499–2527.
25 World Health Organization, What are the WHO Air Quality Guidelines?, **https://www.who.int/news-room/feature-stories/detail/what-are-the-who-air-quality-guidelines**, accessed: 2025-04-09.
26 R. G. Ryan, E. A. Marais, E. Gershenson-Smith, R. Ramsay, J.-P. Muller, J.-L. Tirpitz and U. Frieß, *Atmos. Chem. Phys.*, 2023, **23**, 7121–7139.
27 H. F. Dacre, A. H. Mortimer and L. S. Neal, *Environ. Res. Lett.*, 2020, **15**, 104089.
28 C. A. M. S. (CAMS), CAMS v6.2 Global and European Emission Inventories: Documentation of CAMS v6.2 Global and European Emission Inventories, 2023, **https://atmosphere.copernicus.eu/sites/default/files/publications/CAMS261_2021SC1_D6.1.2-2022_202306_Docu_v1_APPROVED_Ver1.pdf**, accessed: 2025-05-03.
29 N. Christidis, M. McCarthy and P. A. Stott, *Nat. Commun.*, 2020, **11**, 3093.
30 F. Otu-Larbi, C. G. Bolas, V. Ferracci, Z. Staniaszek, R. L. Jones, Y. Malhi, N. R. P. Harris, O. Wild and K. Ashworth, *Glob. Chang. Biol.*, 2020, **26**, 2320–2335.
31 Department for Environment, Food & Rural Affairs and M. Creagh, Government to plant first National Forest in 30 years, 2025, **https://www.gov.uk/government/news/government-to-plant-first-national-forest-in-30-years**, accessed: 2025-05-25.
32 J. M. Kelly, E. A. Marais, G. Lu, J. Obszynska, M. Mace, J. White and R. J. Leigh, *City Environ. Interact.*, 2023, **18**, 100100.
33 M. Vieno, M. R. Heal, S. Hallsworth, D. Famulari, R. M. Doherty, A. J. Dore, Y. S. Tang, C. F. Braban, D. Leaver, M. A. Sutton and S. Reis, *Atmos. Chem. Phys.*, 2014, **14**, 8435–8447.
34 C. P. Webber, H. F. Dacre, W. J. Collins and G. Masato, *Atmos. Chem. Phys.*, 2017, **17**, 867–881.
35 D. Carslaw, *Deweather: Remove the influence of weather on air quality data*, 2025.
36 D. Carslaw, *The Openair Book, openair-project.github.io*, 2025.
37 F. J. Kelly and J. C. Fussell, *Atmos. Environ.*, 2012, **60**, 504–526.
38 T. V. Vu, G. B. Stewart, N. Kitwiroon, S. Lim, B. Barratt, F. J. Kelly, R. Thompson, R. B. Smith, M. B. Toledano and S. D. Beevers, *Build. Sci.*, 2022, **222**, 109359.
39 R. R. Draxler and G. Hess, *NOAA Technical Memorandum ERL ARL-224*, 1997, **https://repository.library.noaa.gov**.
40 R. R. Draxler and G. Hess, *Aust. Meteorol. Mag.*, 1998, **47**, 295–308.
41 R. Draxler, *Memo. ERL ARL-230*, NOAA Air Resources Laboratory, Silver Spring, MD, 1999.
42 A. F. Stein, R. R. Draxler, G. D. Rolph, B. J. B. Stunder, M. D. Cohen and F. Ngan, *Bull. Am. Meteorol. Soc.*, 2015, **96**, 2059–2077.

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